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Evaluating the impacts of land management and climate variability on crop production and nitrate export across the Upper Mississippi Basin

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[1] The increased use of nitrogen (N) fertilizers in the Mississippi Basin since the 1950s is partially responsible for an increase in crop production, but also a massive increase in nitrate export by the Mississippi River. We used the IBIS terrestrial ecosystem model, including new maize and soybean submodels, and the HYDRA hydrological transport model to investigate the role of climate variability, land cover and N-fertilizer application on crop yield, N cycling and nitrate export in the Upper Mississippi Basin from 1974–1994. Simulated annual mean maize and soybean yields were both within 20% of USDA historical estimates in over 80% of the crop-growing counties. There was also strong agreement between simulated and USGS estimated annual nitrate export for the Mississippi River at Clinton, Iowa (r^2 = 0.81), the outlet of the basin, and the Minnesota River at Jordan, Minnesota (r^2 = 0.78). The model also indicated a 30% increase in N-fertilizer application across the basin would have caused only a 4% increase in mean maize yield, but a 53% increase in mean dissolved inorganic nitrogen (DIN) leaching, while a 30% decrease in N-fertilizer application would have caused a 10% decrease in maize yield, but a 37% decrease in DIN leaching. At higher levels of N-fertilizer usage, nitrate export becomes increasingly sensitive to the hydrologic conditions, particularly when there is ample residual N in the soil. Therefore any effort to reduce nitrate export without significantly affecting crop yields would have to account for previous soil-N conditions and climate variability. *INDEX TERMS:* 1871 Hydrology: Surface water quality; 4805 Oceanography: Biological and Chemical: Biogeochemical cycles (1615); 4842 Oceanography: Biological and Chemical: Modeling; 4845 Oceanography: Biological and Chemical: Nutrients and nutrient cycling; *KEYWORDS:* nitrogen, Mississippi River, agriculture, crop yield, nitrate flux, aquatic biogeochemistry

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1. Introduction

[2] Since the 1950s, agriculture and food production have become increasingly dependent on the application of nitrogen (N) fertilizers. The 300% increase in average U.S. maize yields over the past half-century (USDA National Agricultural Statistics Service, available at http://www.usda.gov/nass) (hereinafter referred to as USDA web page, 2001) has been accompanied by an almost 20-fold increase in N-fertilizer use [*Goolsby et al.*, 2000]. A substantial proportion of applied N-fertilizer is not utilized by plants, and leaches to surface waters [*Frink et al.*, 1999]. The leached nitrogen represents a significant economic loss and a serious threat to human health and both the freshwater and marine environment. The problem is particularly acute in the Mississippi River Basin, the world's third largest river basin and home to a $100 billion annual agricultural economy [*Goolsby et al.*, 1999]. A tripling in nitrate (NO_3-) export by the Mississippi River since the 1950s, largely due to increased application of N-fertilizers, has been blamed for an increase in the severity and extent of bottom water hypoxia in the Gulf of Mexico [*Turner and Rabalais*, 1994; *Rabalais et al.*, 1996]. U.S. federal and state negotiators have recently proposed reducing N export by the Mississippi in hopes of reducing the severity and extent of the Gulf of Mexico "dead zone" [*Showstack*, 2000].

[3] The challenge of reducing nitrate export while sustaining crop production is complicated by climate variability. Previous research has suggested that climate variability, as well as the increased dependency on N-fertilizers, is partially responsible for the increased variability in crop production and nitrate flux since the 1980s [*Goolsby et al.*, 2000]. *Donner et al.* [2002] suggested that an increase in precipita-

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tion and runoff could be responsible for over half of the increase in nitrate export from the Upper Mississippi Basin from 1974–1994, by extracting residual N stored in the soil system during dry years. Several researchers have warned that as the soil and groundwater system becomes increasingly saturated with N, the loading to surface water will become increasingly sensitive to changes in hydrology [*Carey et al.*, 1999; *Goolsby et al.*, 2000; *Donner et al*, 2002.]. Potential changes in mean climate and climate variability therefore pose a legitimate threat to crop yields, nitrogen levels in groundwater and the health of aquatic ecosystems.

[4] Previous research investigating the impact of climate and management practices on agricultural landscapes has either been limited to studying the response of selected variables over a large region or studying individual fields or watersheds. A number of large-scale models like EPIC and CERES have been used to investigate the impact of climate and management on U.S agriculture [*Rosenberg et al.*, 1992; *Easterling et al.*, 1993; *Brown and Rosenberg*, 1997; *Easterling et al.*, 1998; *Mearns et al.*, 1999; *Brown et al.*, 2000; *Southworth et al.*, 2000]. However, many existing crop models lack mechanistic representation of physiological (i.e., plant photosynthesis and stomatal conductance) and physical processes (i.e., water, energy, N, and C balance) [*Boote et al.*, 1996], require a multitude of cultivar-specific data (e.g., genetic coefficients), or are incapable of continuous simulation of the soil-plant-atmosphere system [*Zhao et al.*, 2000]. Conversely, a number of combined field and modeling studies have examined the response of both nitrate leaching and crop yields within individual fields to variability in climate and agricultural management [*Boote et al.*, 1996; *Pang et al.*, 1998; *Klocke et al.*, 1999; *Owens et al.*, 2000; *Zhao et al.*, 2000; *Sogbedji et al.*, 2001; *Randall and Mulla*, 2001]. However, these small-scale modeling tools have not been adapted for the study of large river basins. As a consequence, previous regional assessments have largely focused on the response of crop yields to environmental stress without investigating the effect on drainage, nitrogen leaching or nitrate export in river systems.

[5] This study makes a first attempt to assess the impacts of climate variability and land management on nitrate export and crop yields across a large agricultural region. We used an integrated ecosystem model, that includes mechanistic representations of crop phenology and management, and a hydrological transport model to simulate crop yield, nitrogen cycling and aquatic nitrate transport in the Upper Mississippi Basin (UMB) from 1974–1994 (Figure 1). Several sensitivity studies were performed to quantify the effect of varied N-fertilizer use in maize on crop yield and nitrate export across the basin during the past several decades. We hypothesize that a nonlinear relationship exists between N-fertilizer application, crop yields and nitrate export, due to the impact of climate variability, build-up of nitrogen in soils, and the finite response of crop growth to excess available soil nitrogen.

2. Methodology

[6] We used the IBIS terrestrial ecosystem model [*Foley et al.*, 1996; *Kucharik et al.*, 2000] and the HYDRA hydrological transport model [*Coe*, 1998, 2000; *Donner et al.*, 2002] to investigate the role of climate variability, land cover and N-fertilizer application on maize and soybean yield, nitrogen cycling and nitrate export in the UMB. The Integrated BIosphere Simulator (IBIS) simulates crop yields, the water budget and N cycling, including nitrate leaching to the aquatic system, using historical climate forcing for the period 1945–1994. The HYDrological Routing Algorithm (HYDRA) uses the runoff and nitrate leaching simulated by IBIS to simulate river discharge and nitrate export for the period 1974–1994. The first objective of this study was to validate model output against USDA estimates of historical crop yield, literature estimates of individual components of the nitrogen cycle and U.S. Geological Survey (USGS) estimates of river nitrate export throughout the UMB from 1974–1994 (USGS Hypoxia in the Gulf of Mexico Website, 2002, available at http://co.water.usgs.gov/hypoxia/html/nutrients.html). The second objective was to use the IBIS/HYDRA modeling system to assess the impacts of a 30% change in historical N-fertilizer application on maize yields and nitrate export from the basin.

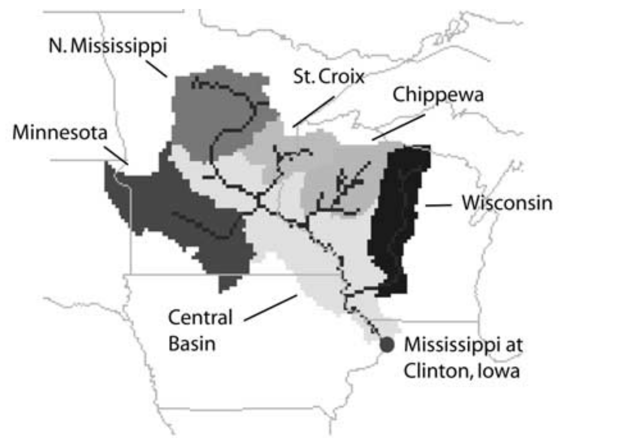


Figure 1. Map of the Upper Mississippi River Basin and the five subbasins examined in this study. This map of the simulated Mississippi Basin was generated from topographic data and manually corrected river directions. The simulated area of each basin is within 10% of observations [*Goolsby et al.*, 1999].

2.1. IBIS Description

[7] IBIS is a dynamic terrestrial ecosystem model that simulates a wide range of phenomena, including land surface processes (energy, water, and momentum exchange); canopy physiology (canopy photosynthesis and conductance); vegetation phenology (budburst and senescence); long-term ecosystem dynamics (vegetation growth, and carbon cycling) and soil biogeochemistry (flow of nitrogen and carbon through vegetation, detritus and soil organic matter) [*Foley et al.*, 1996; *Kucharik et al.*, 2000]. These processes are organized in a hierarchical framework and operate at different time steps, ranging from 60 min. to 1 year. This allows for explicit coupling among ecological, biophysical, and physiological processes occurring on different time scales. IBIS uses climate forcing and basic

physical principles to explicitly simulate the time-transient surface energy and water budget, including surface and subsurface runoff. The model uses a multi-layer formulation of soil to simulate the diurnal and seasonal variations of heat and moisture in the top 4 m of the soil [*Kucharik et al.*, 2000]; in this study, the six soil layers in IBIS were assigned thicknesses of 0.10, 0.15, 0.25, 0.50, 0.50, and 2.5 m. At any time step, each layer is described in terms of soil temperature, volumetric water content, ice content [*Pollard and Thompson*, 1995; *Foley et al.*, 1996], and immobile (e.g., bound to soil aggregates) and mobile (dissolved) inorganic nitrogen. IBIS has been extensively tested and applied toward understanding water and carbon cycling, and vegetation structure in natural grassland and forest ecosystems [*Delire and Foley*, 1999; *Lenters et al.*, 2000; *Kucharik et al.*, 2000, 2001].

[8] The model was recently adapted to simulate both C_3 (e.g., soybean) and C_4 (e.g., maize) crop ecosystems, including terrestrial N cycling, using the logic from several well-documented crop models [*Kucharik and Brye*, 2003]. One might wonder why another model is needed when several other well-tested crop simulation models such as CERES-Maize [*Jones and Kiniry*, 1986; *Mearns et al.*, 1999], EPIC [*Sharpley and Williams*, 1990; *Rosenberg et al.*, 1992; *Easterling et al.*, 1996; *Mearns et al.*, 1999], EPICphase [*Cabelguenne et al.*, 1999], SOYGRO [*Egli and Bruening*, 1992; *Jones et al.*, 1988], and GLYCIM [*Acock and Trent*, 1991; *Haskett et al.*, 1995, 1997] already exist. The goal was to create a generic process-based model based primarily on general differences in C_3 and C_4 plant physiology and phenology that was responsive to management options (e.g., irrigation, fertilizer application, planting date) and environmental stresses (e.g., water and N limitations). This would enable of the simultaneous interactions between climate, land management, soils, crop growth, C and N cycles, and leaching of agricultural chemicals (e.g., nitrate) across the globe, at various scales.

[9] Our modeling approach has taken advantage of the mechanistic nature of IBIS, limiting the number of constants that control crop growth and behavior. Many existing crop models rely on numerous empirical parameters that require adjustment depending on species, hybrid, and geographic location. We have initially constructed and validated the model for maize and soybean, two of the dominant crops in the U.S. However, the generic structure of the modeling framework will permit the inclusion of other crop types in the future, without significant changes, and application in other agricultural regions.

[10] The new IBIS crop component simulates daily LAI development, yield, harvest index, carbon cycling, drainage, agricultural chemical leaching and concentration, plant water and N uptake, C/N ratio of residue and net N-mineralization (minus denitrification). Algorithms similar to those found in the EPIC and EPIC-PHASE models [*Sharpley and Williams*, 1990; *Cabelguenne et al.*, 1999] control plant phenological stages (emergence, grain fill, senescence), shifts in C and N allocation, and N-fixation by soybeans. See complete description by *Kucharik and Brye* [2003]. Leaf area development is thermal time dependent [*Ritchie and Nesmith*, 1991] and is parameterized on the basis of logic found in the CERES-Maize [*Jones and Kiniry*, 1986] model. Nitrogen stress is imposed on plant growth based on leaf N content, which affects the maximum plant photosynthetic capacity (V_{max}) and the C:N ratio in plant residue. The model incorporates a variety of management decisions, including fertilizer application, planting density, irrigation strategy, and planting date.

[11] The nitrogen cycling approach for maize and soybeans has initially been evaluated using data and modeling parameters reported by *Muchow* [1994], *Muchow and Sinclair* [1995], and *Sexton et al.* [1998]. Nitrogen originates from atmospheric deposition, fertilizer application, fixation (by soybeans), mineralization of organic matter, and is removed through vegetation uptake, soil denitrification and leaching to the aquatic system. Each soil layer contains pools of soil organic nitrogen (SON), soil inorganic nitrogen (SIN) and dissolved inorganic nitrogen (DIN). The immobile SIN and mobile DIN pools are kept in dynamic equilibrium, with 10% of total inorganic nitrogen maintained in the DIN pool at each time step. A mechanistic agrochemical-leaching module determines DIN leaching from the soil via subsurface drainage at 1.5 m depth. It is assumed that 95% of leached DIN is in the form of NO_3-, as was observed in field experiments in Wisconsin [*Brye*, 1999].

[12] The simulated N cycle in IBIS for natural ecosystems (e.g., grasses and forests) is less dynamic and complex than for agroecosystems. The sources of nitrogen are the same as in agroecosystem simulation, less inputs of N-fertilizer. However, the photosynthetic capacity of natural vegetation is not controlled by leaf N content. Trees and grasses are assumed to have a plentiful supply of inorganic N for growth at all times, thus N stress is not considered. Instead, plant N uptake is controlled by demand through assimilated carbon (photosynthesis) and the fixed C/N ratios of growing vegetative components (e.g., leaves, roots, and wood). Because the C/N ratios of litterfall in IBIS directly affect C- and N-mineralization (decomposition rates), N-mineralization and plant N demand would be in approximate balance if constant climatic conditions were present. The ratio of NO_3- to DIN in drainage is assumed to be 35% for grassland systems, as was observed in field experiments in Wisconsin [*Brye*, 1999], and 50% in forested systems, based on a findings from a variety of field studies [*Reckhow et al.*, 1980].

2.2. HYDRA Description

[13] We used the HYDRA to simulate river discharge, nitrate export and in-stream nitrate loss in the Upper Mississippi river system at 5' spatial resolution ($\sim$7 $\times$ 9 km). HYDRA is a hydrological transport model that simulates the time-varying flow and storage of water and solutes in terrestrial hydrological systems, including rivers, wetlands, lakes, and human-made reservoirs [*Coe*, 1998, 2000; *Donner et al.*, 2002]. IBIS and HYDRA are linked through a shared water balance; IBIS supplies surface runoff, subsurface runoff and nitrate leaching (at a soil depth of 1.5 m) to HYDRA for transport across the land surface. The models have been extensively tested and applied together to biophysical and hydrological problems at large temporal and spatial scales [*Kucharik et al.*, 2000;

Lenters et al., 2000], and were recently used to examine the role of hydrology in nitrate export by the Mississippi River since the 1950s [*Donner et al.*, 2002].

[14] HYDRA derives river paths and potential lake and wetland volumes from a digital elevation model (Terrain Base, from the National Oceanic and Atmospheric Administration, National Geophysical Data Center). The physical land surface of HYDRA is coupled to a linear reservoir model to simulate the discharge and nitrate flux of river systems and the spatial distribution (and volume) of large lakes and wetland complexes. River discharge, nitrate flux and surface water volume are determined hourly from upstream inputs, local surface and subsurface runoff (from IBIS), local nitrate leaching (from IBIS), point source nitrate inputs, precipitation (from climate data), evaporation from water surfaces (estimated by a simple energy balance model) and river velocity (based on topography). We assumed point source (industrial, municipal) inputs of nitrate to be zero in this study, since they likely represent less than 1% of the total nitrogen inputs to the UMB [*Goolsby et al.*, 1999].

[15] Nitrate is treated like a semi-conservative tracer with the linear reservoir model in HYDRA [*Donner et al.*, 2002]. As in other studies, we assume that benthic denitrification is the only significant nitrate removal process at this scale [*Seitzinger*, 1988; *Seitzinger and Kroeze*, 1998; *Donner et al.*, 2002]. The net change in nitrate mass due to in-stream transformations is negligible, since net inputs of nitrate from the nitrification are insignificant in most rivers in the Mississippi Basin [*Antweiler and Taylor*, 1995; *Peterson et al.*, 2001]. Removal due to sedimentation is small at this scale, since most of DIN accumulated in stream bottom is eventually returned to the river [*Peterson et al.*, 2001]; net loss of nitrate due to biological uptake is also likely to be small [*Battaglin et al.*, 2001], since ammonium is energetically preferable to aquatic plants [*Peterson et al.*, 2001]. Benthic denitrification is determined in HYDRA from water temperature, nitrate availability and water renewal time. In the previous application of HYDRA to the Mississippi Basin, *Donner et al.* [2002] found about 18% of nitrate leached to rivers and streams in the Mississippi Basin is lost via denitrification in sediments, within the 5–20% loss rate predicted by other researchers [*Howarth et al.*, 1996; *Goolsby et al.*, 1999]. Simulated denitrification rates for both cultivated and predominately forested subbasins range also fell within the reported range [*Billen et al.*, 1991]. Because of data limitations, we do not calculate the impact of near-stream riparian zones or isolate denitrification on the floodplain from total river denitrification. However, the model structure permits incorporation of these processes in the future as more data becomes available.

2.3. Model Input Data

2.3.1. Climate and Soils Data

[16] In this study, long-term monthly mean climate data at $0.5^\circ \times 0.5^\circ$ resolution for the period 1940–1995 were obtained from the Climate Research Unit (CRU) of the University of East Anglia [*New et al.*, 2000] and National Center for Environmental Prediction (NCEP) climate reanalysis for the period 1958–1995. In this study, the daily variability for each meteorological variable from the NCEP climate reanalysis were combined with the monthly values from the CRU-05 data set for the period 1958–1995. This provided realistic representation of daily weather events, important to regional crop simulations, while preserving the monthly values anomaly data from the CRU-05 data set. Previous research has questioned the accuracy of the raw values of several meteorological variables in the NCEP daily climate reanalysis [*Lenters et al.*, 2000; *Donner et al.*, 2002]. Hourly average values of air temperature, precipitation, relative humidity, solar radiation, and wind speed were derived using the weather generator within IBIS (WGEN [*Richardson and Wright*, 1984]), which implements monthly mean climatology.

[17] Soil texture as a function of soil depth and layer structure used in IBIS was derived from the Pennsylvania State University Earth System Science Center's CONUS dataset [*Miller and White*, 1998], based on the USDA State Soil Geographic Database. The 30 arcsecond resolution data set was aggregated to 0.5° resolution and used to determine to obtain dominant soil type. Soil physical and hydraulic properties from *Campbell and Norman* [1998] and *Rawls et al.* [1992] were assigned to each soil layer based on a classification of the soil texture into one of 11 major categories (e.g., sandy loam, clay loam, silt loam, etc.)

2.3.2. Land Cover Data

[18] We have classified the land cover in the Mississippi Basin as either permanent maize, permanent soybean or natural vegetation, according to the classification scheme used in IBIS (Table 1). Maize and soybeans are the dominant crops in the UMB, planted on over 75% of the permanent croplands in Minnesota and Wisconsin [*Lander and Moffitt*, 1996]; maize also receives the vast majority of the N-fertilizer applied [*Goolsby et al.*, 1999]. *Donner* [in press] determined the 1992 fractional cover of maize and soybean at a $5' \times 5'$ spatial resolution in the United States by synthesizing the planted area in each county (USDA web page, 2001) with satellite-derived fractional total croplands dataset of *Ramankutty and Foley* [1999]. We used this 1992 fractional maize and soybean cover data for the UMB (Figure 2) for the entire 1974–1994 period of our analysis. Although there was an expansion of soybean cultivation over the period, including a 16,000 km^2 increase in the state of Minnesota since 1970 [*Donner*, in press], it represents a very small portion of our study region. Since the objective of this study was to assess the sensitivity of the basin to variability in climate and changes in total N-fertilizer application, we chose to neglect the impact of changes in land cover.

[19] Natural vegetation type was derived from the 1 km DISCover land cover data set [*Loveland and Belward*, 1997]. *Ramankutty and Foley* [1999] aggregated the original 94 land cover classes into 15 biomes and converted the data to 0.5° resolution by selecting the most dominant biome within each 0.5° grid cell. We assumed the land in each grid cell not covered to maize or soybeans is covered by natural vegetation. We have therefore neglected other croplands, pasture lands and urban areas, which constitute a smaller portion of the nitrogen budget of the study area.

2.3.3. Nitrogen Fertilizer Application

[20] There has been an almost 20-fold increase in N-fertilizer application on crops in the United States since

Table 1. Basin Statistics

Basin	Station Location	Area,[a] km^2	Corn, %	Soybean, %
Minnesota River	Jordan, Minn.	43,607	30.1	26.8
Northern Mississippi	Royalton, Minn.	32,651	3.4	0.3
Chippewa River	Durand, Wis.	22,990	6.8	0.7
St. Croix River	St. Croix Falls, Wis.	16,370	3.6	0.7
Wisconsin River	Muscoda, Wis.	25,539	7.9	0.7
Central basin	na	72,468	15.9	4.6
Upper Mississippi	Clinton, Iowa	213,625	14.0	7.3

[a]Simulated basin area.

the end of World War II [*Alexander and Smith*, 1990]. To represent the historical changes in N-fertilizer usage, we determined annual application rates for maize and soybean using a combination of state-level fertilizer application rate data for each crop and the historical trend in total U.S. N-fertilizer use (Figure 3). The annual fertilizer application rates for 1964–1991 were determined from crop area-weighted averages of the reported historical N-fertilizer application rates for each state in the Mississippi Basin [*USDA*, 1994]. The trend in annual N-fertilizer usage on all crops in the United States from 1945–1985 [*Alexander and Smith*, 1990] was used to "hindcast" the fertilizer application rates for the individual crops types from 1945–1963. The trend for maize and soybean from 1945 to the present was subsequently smoothed using curvilinear and linear regression respectively, assuming no significant change in the rate of application since 1990 [*Frink et al.*, 1999].

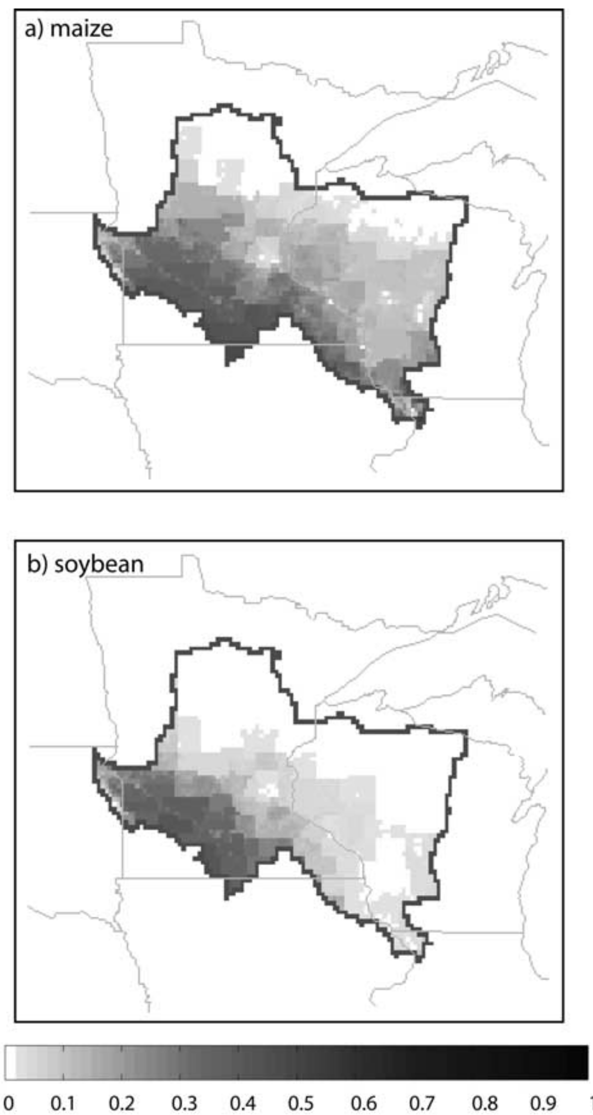


Figure 2. Fractional coverage of (a) maize and (b) soybean at $5' \times 5'$ spatial resolution for the Upper Mississippi Basin (upstream of Clinton, Iowa).

[21] The resulting time series reflects the sharp increase in maize N-fertilizer application during the 1960s and the relatively low level of fertilizer application on N-fixing soybeans. Unfortunately, since there is no consensus on annual maize N-fertilizer application rates in this region, the rates used here differs from that of other modeling and field studies. The annual maize fertilizer rate (150 kg N ha^{-1} yr^{-1} in 1991) in this study is greater than the USDA estimated rates for Wisconsin (101 kg N ha^{-1}) and Minnesota (126 kg N ha^{-1}), but lower than rates (180 kg N ha^{-1}) commonly used by farmers in Wisconsin [*Shepard*, 2000] and in other field and modeling studies [*Sogbedji et al.*, 2001; *Kucharik and Brye*, 2003].

[22] We have intentionally ignored any existing spatial variability in application rates, N-fertilizer applied to other crops and changes in land cover, to focus on the role of

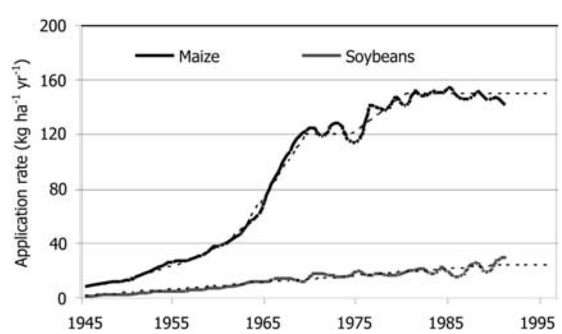


Figure 3. Estimated U.S. nitrogen fertilizer application on maize and soybean from 1945–1995. The average annual application rates were determined from 1964–1991 state-level fertilizer use data for maize and soybean [*USDA*, 1994] and the trend in total U.S. nitrogen fertilizer use from 1945–1985 [*Alexander and Smith*, 1990]. The dashed lines are the result of statistical smoothing, assuming no significant change since 1990.

climate and total N-fertilizer application on maize yield and nitrate export for the contemporary biosphere. Therefore, the total fertilizer input to the model is not equal to other historical estimates of fertilizer use in this region. The total fertilizer application to maize and soybeans (e.g., application rate multiplied by crop area) to the UMB in the model in 1991 is 15% lower than estimates made from county-level fertilizer sales data for the United States [*Battaglin and Goolsby*, 1996]. Conversely, the simulated input in 1974 is 26% greater than estimates from sales data [*Alexander and Smith*, 1990], largely because we employed 1992 cropland cover, which features a larger area of soybeans and maize, throughout the 1974–1994 period.

3. Description of Experiments

3.1. IBIS Simulations

[23] IBIS simulations were performed on a $0.5^\circ \times 0.5^\circ$ terrestrial grid across the study region with an hourly time step. The model was subjected to a 200-year spin up period (1745–1944) during which potential vegetation for the region (e.g., vegetation that could exist without human intervention), was allowed to grow and compete for light and water at each grid cell to establish an equilibrium state for soil biogeochemistry [*Kucharik et al.*, 2000, 2001]. During this period, the 30-year mean monthly climate data from the CRU data set (1961–1990) was used to provide climate driver data.

[24] A series of separate IBIS simulations were subsequently conducted over the period 1945–1994. The first simulation used the vegetation dynamics within IBIS to characterize the potential vegetation patterns across the region in the absence of agriculture. From 1945–1957, CRU monthly mean anomaly data was used to drive the model. Starting in 1958, daily anomalies from the NCEP data set were combined with CRU monthly mean anomalies as part of the statistical combination described above. This provided a 37-year climate record (1958–1994) that was recycled six times to allow the vegetation dynamics and the potential vegetation distribution to reach equilibrium (consistent with the contemporary biosphere across the UMB). Results in this study are reported for the last cycle of climate used.

[25] In the crop simulations, IBIS simulated soil and vegetation conditions representative of 1950, derived from the aforementioned natural vegetation simulation, were used as the initial starting point. Natural vegetation was replaced by either maize or soybean in each grid cell in 1950, and replanted each year. Between 1950 and 1957, CRU monthly mean data was used to drive model simulations. Similar to the natural vegetation run, the NCEP/CRU combined climate data set for 1958–1994 was used for cropping systems during that time period. Planting dates were determined from temperature thresholds (e.g., 10-day average soil temperatures and minimum air temperature); no crop rotations or irrigation were employed. Nitrogen fertilizer was applied each year at the time of planting as a pulse input, and losses of fertilizer due to volatilization were ignored. Two additional maize simulations were conducted to explore the role of N-fertilizer inputs on historical crop yield and nitrate export. In the additional simulations, the historical N-fertilizer application rate (Figure 3) for 1970–1994 was reduced by 30% and increased by 30% respectively. The crop yield, N budget and water budget results from the period 1974–1994 were analyzed in this study.

3.2. HYDRA Simulations

[26] The IBIS simulated surface runoff, subsurface runoff and nitrate leaching below 1.5 m served as inputs to HYDRA. Simulated daily runoff and nitrate inputs from the various simulations were integrated with the fractional crop cover data in HYDRA to estimate nitrate export and river discharge in the Upper Mississippi river system at 5′ resolution from 1970–1994. Additional HYDRA simulations preformed assuming the UMB were covered entirely by maize, soybeans or natural vegetation, to assess the contribution of the individual land cover classes. Separate HYDRA simulations were conducted using runoff and nitrate leaching from each of the three IBIS maize N-fertilizer simulations. USGS observations of annual river discharge and USGS estimates of annual nitrate export from 1974–1994 (USGS Hypoxia in Gulf of Mexico Website, 2002) for the Upper Mississippi at Clinton, Iowa and five major tributaries (Wisconsin, Minnesota, St. Croix, Chippewa and northern portion of the Mississippi) were used to validate HYDRA simulated river discharge and nitrate export.

4. Results

[27] In section 4.1, we examine the maize and soybean yield in the UMB given historical fertilizer application, validating crop yield against USDA data. In section 4.2, we examine the nitrogen budget of the UMB, validating nitrogen budget terms against published estimates. In section 4.3, we validate the nitrate export by the Upper Mississippi at Clinton, IA and five major tributaries against USGS estimates from 1974–1994. Lastly, in section 4.4, we examine the sensitivity of crop yields and nitrate export across the UMB to a 30% change in N-fertilizer application.

4.1. Crop Yields Across the Upper Mississippi Basin

[28] We evaluated the IBIS crop models over the UMB by comparing simulated crop yield with USDA county-level data for maize (for grain) and soybeans (USDA web page, 2001). In the previous modeling study, *Kucharik and Brye* [2003] found that IBIS adequately simulated the impact of interannual weather variability and varied N-fertilizer usage on the water, carbon and nitrogen cycling in an agricultural field in southern Wisconsin. Such detailed evaluation of the models across the entire UMB is impossible, given the lack of widespread, regional datasets of crop variables. Crop yield, however, is an ideal variable for validation because it is both a direct measure of agricultural productivity and a reflection of water and N uptake by crops, which ultimately impacts nitrate flux to the river system. The 1985–1994 period was chosen for analysis because IBIS was originally calibrated to represent 1990s yields, given current N-fertilizer usage and contemporary agricultural "technology."

[29] The simulations captured the general pattern in observed annual maize and soybean yields across the UMB during the 1985–1994 period (Figures 4a and 5a).

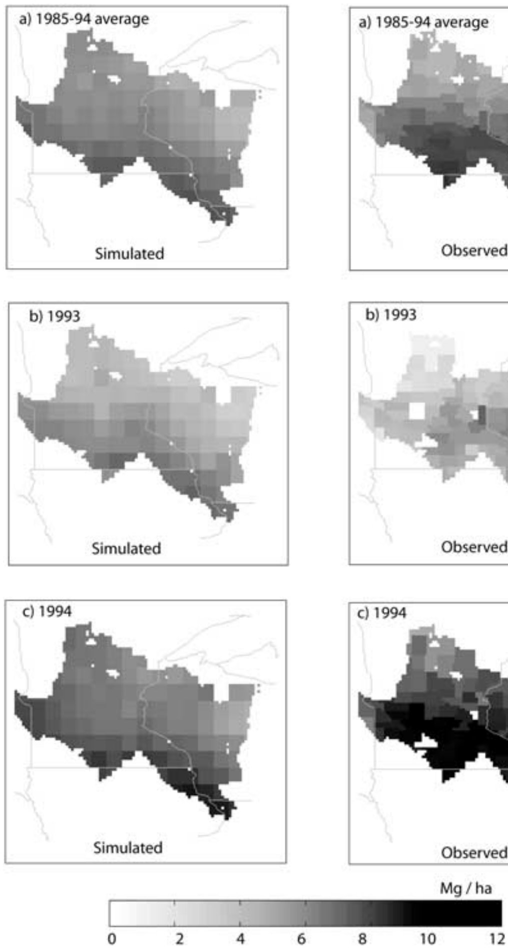


Figure 4. Simulated and observed maize yield (Mg ha^{-1}) for (a) 1985–1994 average, (b) 1993 and (c) 1994. Yield is displayed for each $5' \times 5'$ grid cell with at least 1% soybean cover. Observed maize yield was determined from USDA county-level estimates. See color version of this figure at back of this issue.

There was an underestimate of the heterogeneity in yields, particularly maize, as is to be expected in the application of a generic model with constant agricultural management (e.g., fertilizer application rates) over a large region. There was a negative bias in simulated yields in the highly productive southern portion of the UMB, site of the majority of total crop production, and a positive bias in the less intensively cultivated northern portion of the UMB.

[30] The predictive ability of the modeling system was greater in years featuring a more typical climate. In 1993, an extremely wet year that featured a large spring flood in the southern half of the UMB, there was a large gradient in observed maize yields across the basin (Figures 4b and 5b). IBIS overestimated maize yields in northern Minnesota, as in other years, but also overestimated yield by 20–60% in parts of southwestern Minnesota and Iowa that were adversely affected by flooding. Alternatively, in 1994, an extremely productive year for maize and soybeans, the observed range in yields across the basin was considerably smaller (Figures 4c and 5c). Accordingly, the error in simulated maize and soybean yields was more uniform: the model underestimated maize yields by less than 20% and overestimated soybean yields by less than 30% over much of the UMB.

[31] We also assessed the predictive ability of the model by aggregating the simulated crop yields to the country-level for direct comparison with the USDA data. The resulting histograms of error in annual mean crop yield (Figure 6a) and annual crop yield (Figure 6b) further demonstrate the ability of the modeling system to capture much of the variability in crop yields across a large region. Simulated annual mean maize yield from 1985 to 1994 was within 20% of observations for 94 of the 114 maize-growing counties (82%); soybean yield was within 20% of observations in 73 of the 83 soybean-growing counties (88%). The histogram of percent error in annual crop yield (Figure 6b) reflects the greater prediction error in individual years, seen in Figures 4 and 5. It also suggests a positive bias in both maize and soybean yields across the UMB, although this is a consequence of geography rather than

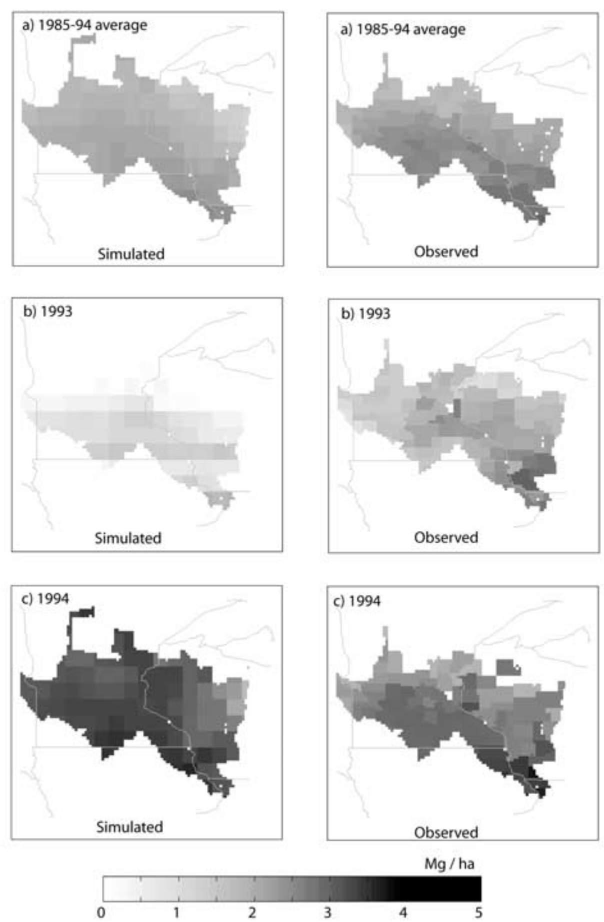


Figure 5. Simulated and observed soybean yield (Mg ha^{-1}) for (a) 1985–1994 average, (b) 1993 and (c) 1994. Yield is displayed for each $5' \times 5'$ grid cell with at least 1% soybean cover. Observed soybean yield was determined from USDA county-level estimates. See color version of this figure at back of this issue.

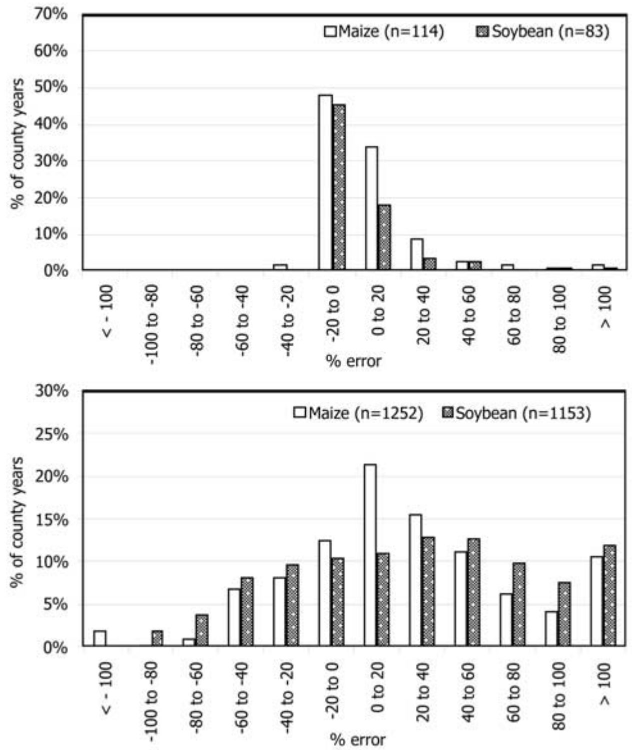


Figure 6. Histogram of percent error in (a) mean annual crop yield and (b) annual crop yield, by county from 1985–1994. Every county in the Upper Mississippi Basin with more than 1% cover of the individual crop were included in the analysis. The percent error is (simulated–observed)/observed * 100.

model function. Crop yields, particularly maize, were overestimated in the large number of northern counties where fewer crops are actually grown. The model actually underestimates total maize production most years due to an underestimate of crop yields in the intensively cultivated counties of southern Minnesota and northern Iowa.

[32] The general agreement between simulations and observations, without accounting for the variation in agricultural management across the UMB in the model, indicates that climate and soils are the primary factors influencing agricultural productivity over a long timeframe. It is unlikely that further model tuning and calibration would minimize error, because it would affect each grid cell in a similar manner. The error in specific regions is most probably a consequence of adopting an average annual rate of fertilizer application for the entire UMB. For example, the negative bias in maize yields in the primary maize-growing regions may be due to adopting a lower application rate (150 kg N ha^{-1}) in this study than in the previous IBIS study (180 kg N ha^{-1}) or in reality for many farms [*Shepard*, 2000]. Since plant photosynthetic capacity in IBIS (e.g., V_{max}) is sensitive to leaf N-content, a slightly lower N-fertilizer application would result in decreased yield. The difference of 30 kg N ha^{-1} in assumed N-fertilizer application between the two studies therefore might partially explain the negative bias in simulated maize yield in the Wisconsin, Central, and Minnesota subbasins. The negative bias in simulated soybean yields, however, is likely not caused a discrepancy in N-fertilizer data, since little fertilizer (<25 kg N ha^{-1}) is typically applied to soybeans [*USDA*, 1994].

[33] It is difficult to draw definitive conclusions about model capabilities in the northern part of the UMB, because the region is not heavily cropped (Figure 2). The positive bias in simulated maize yields may again be a result of assuming constant management across the UMB; agricultural fields in lightly cropped regions tend to receive less fertilizer and not be as intensively managed. The high simulated crop yields suggest either there is potential for higher crop productivity in that region or the soils and climate data used to drive the model may not have been representative of the region. These are potentially important discoveries to explore further in future studies on crop responses to changes in regional climate.

[34] It is important to recognize that the USDA county-level data represent a composite average of the individual reports from farmers across each county, integrating the effects of independent farmer decision-making (e.g., management practices), soil type, topography and mesoscale meteorology on crop productivity. In the case of IBIS simulations, however, yield and other output quantities are simulated from average conditions (e.g., N-fertilizer use, climate, soils) within a $0.5° \times 0.5°$ grid cell. Therefore, there are some potential limitations to generating realistic average yields for a particular county with IBIS. For example, crops may be grown only in a specific part of the county, particularly in areas with low fractional crop cover (Figure 2), where the soil is markedly different than the "representative" soil type used in $0.5° \times 0.5°$ degree IBIS simulations. Previous sensitivity studies have also shown that the weather and N-fertilizer usage, which in reality may be highly variable at the sub-grid-scale, are most influential on crop yield. In addition, IBIS does not account for insects, disease, or weeds, although these factors are likely more important in applications at the scale of individual farms or small watersheds. We envision that future studies will need to assess the model at much higher spatial resolution (e.g., $5' \times 5'$), so that the finer spatial heterogeneity in soils, N-fertilizer management, planting date, and climate information can be implemented.

4.2. Terrestrial Nitrogen Budget

[35] The simulated terrestrial nitrogen budget for the UMB is dominated each year by inputs from net soil mineralization and fertilizer application and uptake by plants (Table 2). The simulated mean annual budget terms are within the range of related literature estimates, despite omitting some smaller sources and sinks of N from this study. The annual input of N from atmospheric deposition, fertilizer application and mineralization were all within 15% of estimates from the *Goolsby et al.* [1999] Mississippi nitrogen budget study. The inputs from N-fertilizer are expected to be 15% lower than other estimates, due to focusing only on the cultivation of maize and soybean in this study. The simulated input from net soil mineralization is within 1% of the estimate by *Goolsby et al.* [1999], but

Table 2. Simulated Nitrogen Budget From 1974–1994[a]

Component	Annual Mean, kg ha^{-1} yr^{-1}	Contribution, %
	Inputs	
Atmospheric deposition	4.0 ± 0.3	4.8
Fertilizer	32.1 ± 1.5	27.8
Mineralization	51.5 ± 5.2	62.0
Fixation	4.4 ± 1.7	5.3
	Outputs	
Plant uptake	69.8 ± 4.4	84.1
Crops	31.4 ± 3.2	37.8
Natural vegetation	38.4 ± 1.9	46.3
Leaching (DIN)	6.9 ± 3.2	8.3
Residual	6.3 ± 5.5	7.6

[a]The annual mean represents the basin-wide mean for the 1974–1994 period. The standard deviation of the mean from 1974–1994 is also reported.

less variable across the basin; the simulated net N-mineralization rate is 40% lower in the Minnesota basin, and nearly twice as large in the other primary subbasins. The simulated input of N from fixation is an order of magnitude lower than the estimate from *Goolsby et al.* [1999], due to a lower simulated annual rate of fixation by soybeans (37.1 ± 20.5 kg N ha^{-1} yr^{-1} versus 78 kg N ha^{-1} yr^{-1}) and the exclusion of other N-fixing crops like alfalfa. However, the simulated rate of fixation by soybeans is still within the range of other literature estimates [*Goolsby et al.*, 1999].

[36] Plant uptake accounts for the vast majority of the N consumed annually in the UMB; maize and soybean uptake accounts for almost half the total plant uptake, despite accounting for only 21% of the basin's land cover. For example, the annual N uptake rate for maize ranged from 126 to 189 kg/ha/yr, and accounting for over 80% of N budget in maize lands, similar to observations from field studies [*Davis et al.*, 2000; *Sogbedji et al.*, 2000]. Plant uptake is likely underestimated in the Minnesota River Basin, due to an underestimate of N inputs from fertilizer and mineralization. Since plant photosynthetic capacity is limited by nitrogen availability, an underestimate of inputs could partly explain the underestimate of crop yields in southern part of the UMB.

[37] Nitrogen not taken up by plants was either leached to the river system as DIN or stored in the inorganic-N or organic-N pools in the soil system. The simulated N budget suggests nitrogen has built up in the soils since the mid-1970s, primarily due to fertilizer application on maize. The residual N term was negative only during 1986 and 1993, when high runoff extracted DIN stored in the soil-groundwater system during previous years. The high loss of residual soil N and fertilizer input during these years, particularly during 1993, partly explains the collapse of maize yields across the UMB.

[38] The vast majority of simulated DIN leaching therefore originates in the heavily fertilized maize ecosystems; only 6% originates from forests and grasslands systems, even though they account for 79% of the simulated land area. The simulated mean annual maize leaching rate (38 ± 14 kg N ha^{-1} yr^{-1}) is at the higher end of the range reported in the literature (Table 3). Since the annual rate of nitrate leaching is highly dependent on weather, fertilizer application and soil texture, it is difficult to validate the simulated leaching rates against the results of individual field studies. However, the simulated maize leaching rates follow the expected pattern. For example, the simulated leaching rate ranged from 16.8–47.3 kg N ha^{-1} in grid cells in southern Minnesota during 1980–1994 when 150 kg N ha^{-1} of nitrogen was applied. *Sogbedji et al.* [2000] applied 134 kg N ha^{-1} yr^{-1} to maize in southern Minnesota, and measured leaching losses from maize cultivation of 8.2–22 kg N ha^{-1} yr^{-1} in clay soils and 17–35 kg N ha^{-1} yr^{-1} from a sandy loams. As expected, the leaching rate was characteristically larger in soils with a higher sand fraction. The related study by *Kucharik and Brye* [2003] reported measured nitrate leaching rates in maize of 5.9–102.0 kg N ha^{-1} yr^{-1} on a silt loam soil in southern Wisconsin during 1995–2000. The mean nitrate leaching rate during the period was 57.0 kg N ha^{-1} yr^{-1} for maize receiving 180 kg N ha^{-1} of ammonium nitrate fertilizer at planting.

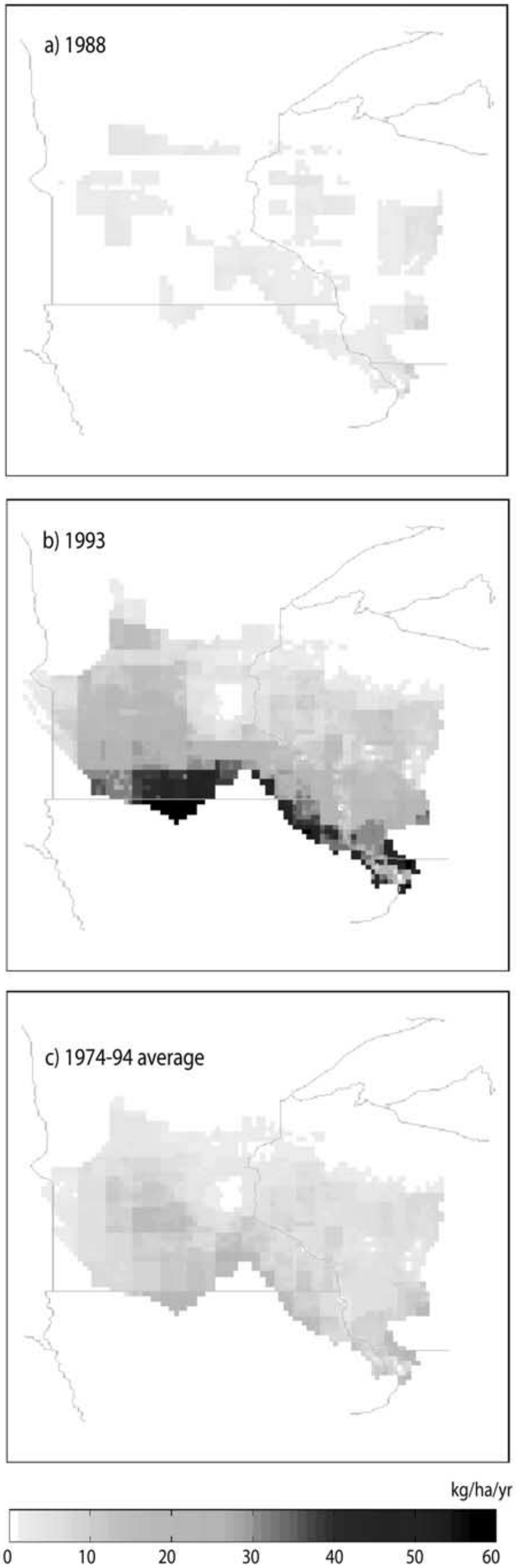
[39] The simulated leaching rates for soybeans are an order of magnitude lower than the maize leaching rate, and lower than most reported rates. The model likely overestimated leaching from maize and underestimated leaching from soybeans likely because each crop was grown continuously, rather than rotated annually among individual fields at the sub-grid-level each year. Nitrate leaching losses from maize grown in rotation with soybean are generally lower than losses from continuous maize cultivation [*Owens et al.*, 2000; *Randall and Mulla*, 2001]. We hope to include crop rotations at the sub-grid-level in future IBIS simulations, to offer a more realistic simulation of modern agricultural practices. Although the difference in maize and soybean leaching rates is exaggerated, the simulated leaching rates still fall within the wide expected range for row crops (2.1–79.6 kg/ha) compiled by *Reckhow et al.* [1980].

[40] The simulated nitrate leaching rates from natural vegetation, particularly deciduous forests and grasslands, are at the high end of the range of leaching rates reported in the literature (Table 3). The simulated nitrate leaching rates

Table 3. Nitrate Leaching by Land Cover Class 1974–1994[a]

Land Cover Type	Area, km^2	Percent of Basin	Simulated, kg ha^{-1} yr^{-1}
Maize	31,602	14.8	38.0 ± 18.4
Soybean	16,509	7.7	1.9 ± 1.0
Temperate/evergreen conifer forest	66,467	31.0	0.8 ± 0.7
Temperate/evergreen deciduous forest	6191	2.9	1.1 ± 1.1
Boreal evergreen forest	73,237	34.2	0.26 ± 0.2
Grassland	20,109	9.4	1.7 ± 0.7

[a]The annual mean represents the basin-wide mean for the 1974–1994 period. The standard deviation of the mean from 1974–1994 is also reported.



of 1.43 kg N ha^{-1} yr^{-1} for temperate/evergreen deciduous forests and 4.53 kg N ha^{-1} yr^{-1} for grasslands are significantly lower than the rates (0.32, 0.4 kg N ha^{-1} yr^{-1}) derived from the literature in Chapter Two [*Donner et al.*, 2002]. A possible explanation for the difference is that a smaller proportion of total soil inorganic-N is in the form of nitrate in natural ecosystems, than in croplands, since no potentially mobile N-fertilizer is applied. This study, however, assumes the same equilibrium constant between DIN and SIN in both croplands and natural vegetation. Another potential shortcoming of our simulations could be the less dynamic N cycling that is applied to natural vegetation.

[41] The dependence of leaching on land cover is demonstrated by the variation in DIN leaching across the study region (Figure 7), which mirrors the variation in fractional maize cover. The model indicated that the Minnesota River basin is responsible for the greatest proportion of leached nitrogen, as estimated by the *Goolsby et al.* [1999]. The model suggests that during wet years, like 1986 and 1993, humid regions with intensive maize cultivation like southeastern Minnesota become "hot spots" of nitrate leaching. *Randall and Mulla* [2001] found that nearly 75% of the nitrate load of the Minnesota River originates from eastern subwatersheds.

4.3. River Nitrate Export

[42] In previous applications to the U.S. water balance [*Lenters et al.*, 2000; *Donner et al.*, 2002], IBIS and HYDRA accurately simulated annual river discharge by the Upper Mississippi river and the major tributaries, providing confidence in the ability of the modeling system to represent the effect of hydrologic processes on nitrate export. In this study, simulated annual river discharge by the Mississippi at Clinton, Iowa is again closely correlated to USGS observations ($r^2 = 0.73$), although the model underestimates discharge by 27% during the 1993 flood year (Figure 8). There are subtle differences in the water balance in this study, due to the simulation of agricultural systems and the incorporation of NCEP daily climate anomalies.

[43] We validate the simulated annual nitrate export by the Mississippi River at Clinton, Iowa (outlet of the basin) and five major tributaries against USGS estimates of annual nitrate export from 1974–1994, determined from point measurements of concentration and a multiple regression model relating concentration to discharge and seasonality [*Goolsby et al.*, 1999]. There is strong agreement ($r^2 = 0.81$) between the variability in simulated and USGS estimated annual nitrate export by the Mississippi River at Clinton, Iowa (Figure 9). The simulation clearly captures the major events over the study period, including the rapid increase in nitrate export during the 1974–1986 period, the drop during

Figure 7. (opposite) Simulated annual dissolved inorganic nitrogen leaching (kg ha^{-1} yr^{-1}) over the Upper Mississippi Basin for (a) 1988, (b) 1993 and (c) 1974–1994 average. Simulated leaching rates from the natural vegetation, soybean and maize control run were integrated with the fractional crop cover data to develop these maps. See color version of this figure at back of this issue.

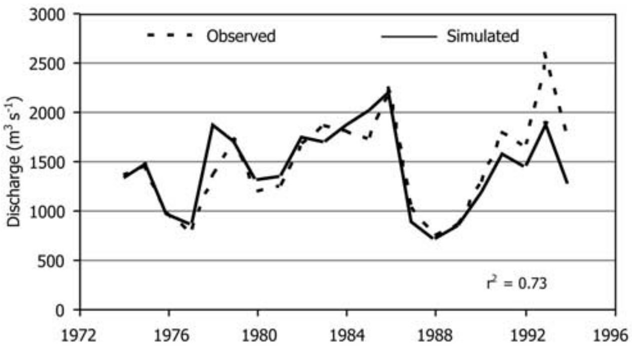


Figure 8. Simulated (dashed line) and observed (solid line) annual river discharge (m^3 s^{-1}) from the Mississippi River at Clinton, Iowa from 1974–1994. The correlation coefficient (r^2) between simulated and observed annual discharge is displayed in the lower right corner.

the 1987–1989 dry period and the peak in export during the 1993 flood year. The correlation between simulated and USGS estimated annual nitrate concentration is slightly weaker (r^2 = 0.59), since simulated concentration incorporates error in both river discharge and nitrate mass.

[44] The model suggests 12–22% of the nitrate leached to the river system is lost annually via benthic denitrification, predominately in smaller streams draining intensively cultivated catchments (Table 4). The aggregate nitrate loss due to denitrification is in accordance with other published estimates for large basins with significant N loading [*Howarth et al.*, 1996]. *Alexander et al.* [2000] predicted 40% annual retention in the UMB, but based on total N, and thus including processes other than denitrification, like deposition of particulate nitrogen. The variation in nitrogen retention due to denitrification in rivers and streams is a key issue for management of the UMB and will be addressed in a future study.

[45] Mean nitrate export for the period is 20% greater than USGS estimates, despite neglecting some alternative N sources, like municipal and industrial point sources, and some sinks, like retention in the riparian zone. But the accurate simulation of interannual variability suggests the role of other N sources and sinks is small, and does not vary substantially on an interannual basis. The high mean simulated nitrate export is therefore likely due to the anticipated overestimate of fertilizer application during the first half of the study. High fertilizer application resulted in greater nitrate leaching from 1974–1986, and contributed to the build-up of N in the soil system, which was flushed out during later high precipitation years like 1986 and 1993. In addition, the model may underestimate plant uptake, evident in the underestimate of crop yields in the heavily cultivated Minnesota basin. Lastly, the model may assume too high a ratio of NO_3- to DIN in subsurface drainage from forests and grasslands where NH_x may often dominate, but also from agricultural lands where NO_3- clearly dominates. The model assumption that 95% of DIN in agricultural fields is leached in the form of NO_3-, based on measurements on agricultural fields in Wisconsin [*Kucharik and Brye*, 2003], may be too high [*Sogbedji et al.*, 2000]. For example, if only 80% of the simulated DIN from agricultural lands leached as NO_3-, the mean annual nitrate export for the period would only be 10% greater than USGS estimates.

[46] In general, the modeling system simulates the expected overall pattern in nitrate yield (nitrate export divided by basin area) across the UMB, but reports less variation among the major subbasins (Figure 10). Nitrate yield is underestimated in the Minnesota Basin and overestimated in the other more forested subbasins. Budget analysis suggests the discrepancy is due to the low variability in simulated mineralization rates mentioned in the previous section, but also due to the exclusion of artificial drainage practices, the exclusion of spatial variability in fertilizer application and the high DIN leaching rate from natural systems. First, as much as half the croplands in the Minnesota River basin are artificially drained, increasing the potential for nitrogen leaching [*Zucker and Brown*, 1998; *Zhao et al.*, 2000]; *Brezonik et al.* [1999] also attributed an underestimate of nitrogen export from the Minnesota River Basin by HUMUS on the model's inability to simulate tile drainage. Second, USDA statistics and field studies suggest fertilizer application rates are roughly 25% greater in Minnesota than in Wisconsin [*USDA*, 1994; *Sogbedji et al.*, 2000]. Lastly, the high nitrate leaching rates from deciduous forests, which cover 31% of the UMB but only

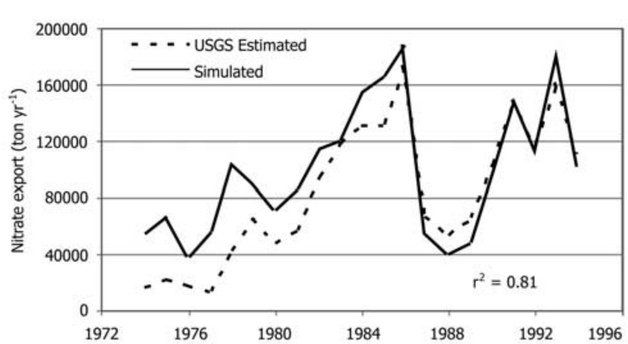


Figure 9. Simulated (dashed line) and USGS estimated (solid line) annual nitrate export (ton yr^{-1}) by the Mississippi River at Clinton, Iowa from 1974–1994. Nitrate export is simulated with historical fertilizer use on maize and soybeans.

Table 4. Simulated Nitrate Loss Due to Benthic Denitrification (1974–1994)[a]

Basin	Cropland, %	Percent Nitrate Removed		
		Annual Mean	Minimum	Maximum
Northern Mississippi	3.7	8.7 ± 1.1	5.5	11.2
St. Croix	4.3	13.1 ± 1.9	10.5	18.6
Chippewa	7.5	10.9 ± 1.5	8.2	13.4
Wisconsin	8.6	14.4 ± 3.0	10.3	21.8
Minnesota	57.8	13.9 ± 3.6	10.4	24.3
Upper Mississippi	21.3	15.8 ± 3.1	12.1	23.0

[a]The annual mean represents the basin-wide mean percent of nitrate removed via benthic denitrification for the 1974–1994 period. The standard deviation of the mean from 1974–1994 is also reported. The minimum (maximum) represents the annual minimum (maximum) basin-wide percent nitrate removed for the 1974–1994 period.

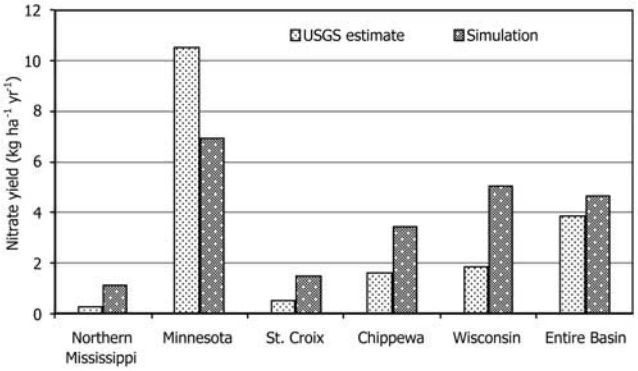


Figure 10. Simulated and USGS estimated mean annual nitrate yield (kg ha^{-1} yr^{-1}) from 1974–1994 for the Upper Mississippi subbasins. The nitrate yield is the nitrate export from the basin divided by the basin area.

11% of the Minnesota basin, largely explains the overestimate of leaching from the predominately forested St. Croix, northern Mississippi and Chippewa subbasins. For example, the difference between the simulated leaching rate for temperate/evergreen deciduous forests (1.43 kg N ha^{-1} yr^{-1}) and the estimate used by *Donner et al.* [2002] (0.32 kg N ha^{-1} yr^{-1}) explains over 40% of the difference in simulated and USGS estimated nitrate export from the St. Croix Basin. It is also likely that the relative contribution of nitrogen from sources ignored in this study, including manure, industry and sewage, is greater in the Minnesota basin, but the total contribution is small in contrast to fertilizer and mineralization [*Goolsby et al.*, 1999].

[47] Despite the underestimate the spatial variability in nitrate yield across the UMB, there is still strong agreement between the interannual variability in simulated and USGS estimated annual nitrate export by the five major tributaries (Figure 11), particularly the Minnesota River at Jordan, Minnesota ($r^2 = 0.78$). The accurate simulation of variability in export from the Minnesota River Basin, the most intensively cultivated part of the UMB (57% maize and soybean), indicates the ability of the modeling system to capture N cycling in croplands. The agreement between simulated and USGS estimated export is weaker in the less cultivated subbasins like the Chippewa due in part to the overestimate of leaching from forests (Figure 11b). However, at those stations, we found stronger agreement between simulated export and USGS estimates of Total N (USGS Hypoxia in the Gulf of Mexico Website, 2002), which includes NH_x, NO_x and organic-N. This further suggests the assumed NO_3-: DIN ratio is too large, particularly in predominately forested catchments. In future applications of this modeling system to large river basins, we hope to differentiate between the various N species in soil water.

4.4. Impacts of Varied N-Fertilizer Use in Maize on Crop Yield and Nitrogen Cycling

4.4.1. Crop Yield

[48] The model suggests that a change in the N-fertilizer use results in a disproportionately smaller change in mean maize yields across the UMB (Figure 12). Reducing the N-fertilizer application by 30% results in a 10% decrease in mean maize yield for the region from 1974–1994. However, increasing the fertilizer application by 30% results in only a 4% increase in yield. It should be noted that the loss in revenue from even a small change in maize yields likely exceeds the profit margins of many individual farms. The impact of a change in fertilizer application on yield depends on a variety of factors, including N availability in the soil and advances in technological breeding, but appears to vary primarily with the hydrologic conditions.

[49] The simulated decrease in yield caused by a decrease in N-fertilizer application is greatest during 1986 (16%), 1977 (14%) and 1993 (12%), three of the years with the greatest precipitation and runoff in the study period. Runoff and nitrate leaching losses are relatively high during those years, so crop growth becomes more nutrient limited. Conversely, the decrease in fertilizer application had the least impact on yield during 1988 (4%) and 1989 (7%), two of the driest years during the study period. In those years, nitrate leaching losses are relatively low, leaving more N available for plant uptake. Crop growth is likely increasingly limited by water availability, so an increase in fertilizer application has little effect on yield in years when weather conditions (e.g., rainfall) are the most limiting factor. Moreover, past studies have clearly shown that the relationship between N-fertilizer application and maize yield is approximated by a rectangular hyperbole

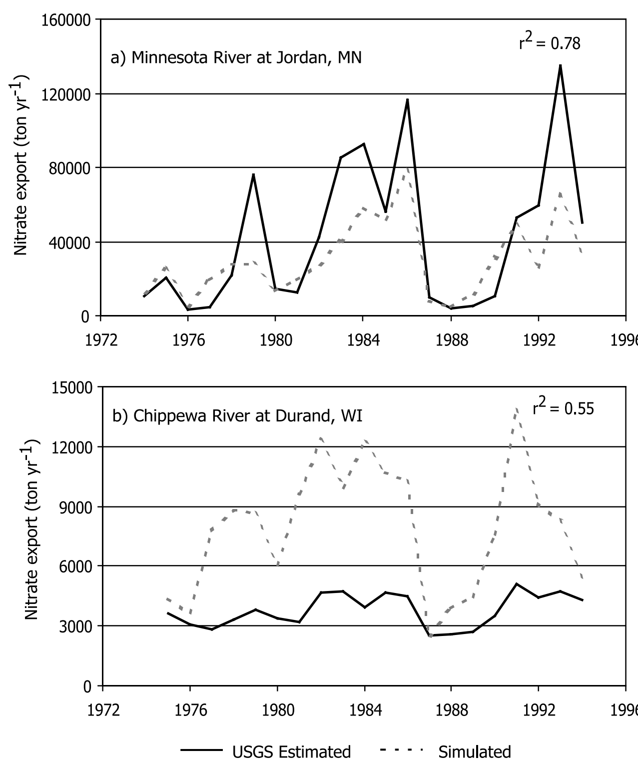


Figure 11. Simulated (dashed line) and USGS estimated (solid line) annual nitrate export (ton yr^{-1}) by the (a) Minnesota River at Jordan, Minnesota, and (b) Chippewa River at Durand, Wisconsin.

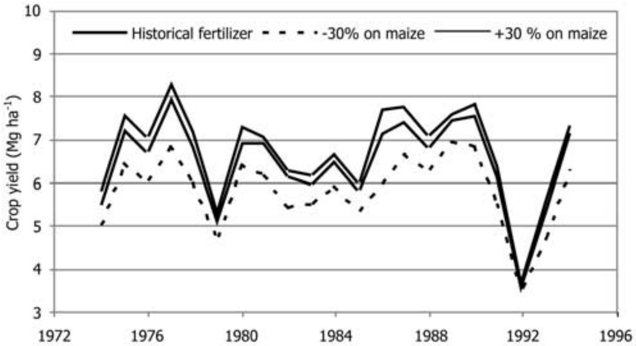


Figure 12. Simulated annual crop yield (Mg ha^{-1}) in the Upper Mississippi Basin from 1974–1994 given historical fertilizer use, a 30% reduction in fertilizer use each year and a 30% increase in fertilizer use each year.

[*Vanotti and Bundy*, 1994a, 1994b]. Thus, in many years when plentiful soil inorganic N is available so plants reach an optimal N level, excess N will not lead to yield increases. Farmers typically apply excess N-fertilizer to help combat leaching losses during spring rains, but if this excess N remains in the soil, it only adds to an increase in SIN in the long-term, and magnifies leaching losses during wet years.

[50] Therefore, hydrologic conditions play a crucial role in the impact of increased N-fertilizer on yields due to the greater storage of N in the soil. The increase in simulated mean maize yield due to the 30% increase in fertilizer application was greatest in 1986 (8%), the wettest year of the study period, as the high fertilizer input compensated for the high leaching losses. But there is an overall decrease in the impact of the 30% increase in fertilizer on simulated yields (significant at the 85% level), likely due the availability of ample residual nitrogen for plant uptake. Although greater fertilizer input still affects yields during extremely wet years like 1993 (5% increase), the impact on yield during even moderately dry years like 1992 (2.5% increase) is quite small. This again illustrates how soil water storage can be the most limiting factor to plant growth. There may be an ample supply of soil inorganic N, but the plant is unable to use it.

4.4.2. River Nitrate Export

[51] The simulations suggest nitrate export is extremely sensitive to changes in fertilizer application, particularly during wet years (Figure 13). The 30% increase in maize fertilizer application has a disproportionately greater effect on export than the 30% decrease, as more fertilizer is applied than required by the crops, resulting in a greater percentage stored in the soil or leached to river system. The impact of a 30% change also magnifies over time, due to the change in soil N storage. By the early 1990s, the 30% increase in fertilizer results in an almost 60% increase in nitrate export, while the 30% decrease results in a 40% decrease in export. The effect of a fertilizer increase is magnified more during wet years; in 1993, the increase in export is equivalent to 83% of the total increase in fertilizer application. These trends indicate that above a threshold level of fertilizer application, nitrate export may increase exponentially, but also that a long-term reduction in fertilizer use could significantly reduce nitrate export in the future.

[52] The sensitivity of nitrate export to precipitation also appears to increase substantially at higher levels of fertilizer application. There is a significant linear relationship between simulated annual nitrate export (at Clinton, Iowa) and annual precipitation across the UMB in each fertilizer scenario (Table 5), as has been noted in a number of other studies on both fertilizer croplands and forested catchments [*Lucey and Goolsby*, 1993; *Creed and Band*, 1998; *Zhao et al.*, 2000]. However, the 30% increase in fertilizer application on maize has a greater impact on the precipitation-export relationship (18,000 ton increase in export with each 100 mm of precipitation) than the 30% decrease in fertilizer (13,200 ton decrease). In addition, the significance of the precipitation-export relationship also decreases with increase in fertilizer application. Therefore, as fertilizer application increases and soil N storage increases the precipitation-export relationship appears to become increasingly nonlinear. The predictability of each precipitation-export relationship increases when the annual fertilizer input is included in the regression equation, which further demonstrates how leaching is largely driven by precipitation but is still limited by N inputs to the soil system.

5. Discussion and Conclusions

[53] In general, the modeling system effectively simulated crop yields, nitrogen leaching and river nitrate export across the Upper Mississippi Basin, given estimated historical changes in fertilizer application on maize and soybeans. The ability to simulate both the carbon (e.g., yield) and nitrogen (e.g., nitrate export) cycles across a large river basin will be key to future studies of agricultural management and environmental change. Further modifications to IBIS and HYDRA, including explicit representation of maize-soybean rotations, inclusion of other crops and differentiation of dissolved N species in soil water and drainage, would enhance simulation of N cycling. The strong relationship between fertilizer application, climate,

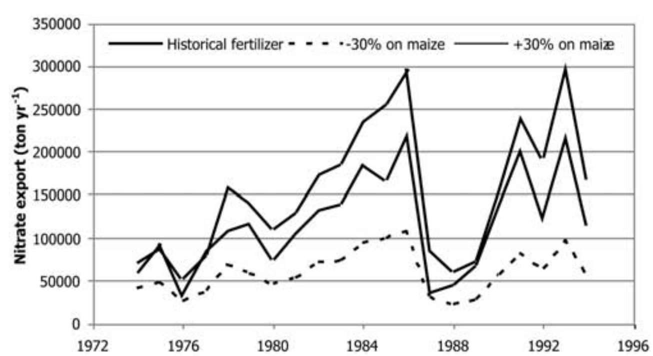


Figure 13. Annual nitrate export (ton yr^{-1}) by the Mississippi River at Clinton, Iowa from 1974–1994 given historical fertilizer use, a 30% reduction in maize fertilizer and a 30% increase in maize fertilizer.

Table 5. Relationship Between Annual Nitrate Export and Precipitation

Scenario	r^2	Slope, ton mm^{-1}	p-Value	r^2, + Fertilizer[a]
USGS estimates	0.26	242	0.0178	n/a
Historical fertilizer	0.47	306	0.0006	0.56
Maize −30%	0.51	174	0.0003	0.54
Maize +30%	0.44	486	0.0011	0.57

[a]Multiple regression with precipitation and annual fertilizer application.

N losses and crop yields demonstrated in this study highlights the potential economic and environmental benefits of precision agriculture.

[54] The ability of the modeling system to simulate the local impact of climate and management on agricultural productivity and environmental health is largely limited by the resolution of both the model and the input data. For example, the integration of high-resolution fertilizer use data would clearly improve simulation of the variability in crop yields and nitrate leaching across a large region. The ideal way to interpret the performance of IBIS in different counties across the UMB would be to perform simulations at much higher resolution. The scale of analysis, however, will depend on the availability of quality high-resolution climate, soils and fertilizer data.

[55] The simulations of temporal variability in crop yields and nitrogen cycling is also limited by the ability of a weather generator (WGEN [*Richardson and Wright*, 1984]) to simulate real weather events from long-term mean climate data. Extreme weather events and strong gradients in precipitation between grid cells, which significantly impact crop production, are smoothed over in large-scale long-term mean monthly climate data. In the previous validation study, IBIS accurately simulated interannual crop yield for an individual agricultural field when forced with hourly micrometeorological data collected on-site [*Kucharik and Brye*, 2003]. But when forced with CRU-05 monthly mean climate and NCEP daily weather events for the field site location, IBIS produced significantly less (30%) variability of crop yield over a 6-year period [*Kucharik*, unpublished data]. The error in county-level annual crop yield in this study (Figure 6b) could be partly a result of the weather generator not describing the extreme events that significantly impact fertilizer use efficiency and crop yield. Weather generator capabilities will likely be explored in a future study.

[56] The accuracy of the model over the whole basin enabled analysis of the impact of broad changes in N-fertilizer application on both crop yield and nitrogen losses to the river system. The multiple fertilizer scenarios indicate the response of maize yield to changes in fertilizer application is small in contrast to the response of DIN leaching and eventual nitrate export from the basin (Figure 14). A 30% increase in fertilizer application causes only a 4% increase in maize yield, but a 53% increase in DIN leaching; a 30% decrease in fertilizer application causes a 10% decrease in maize yield, but a 37% decrease in DIN leaching. The results suggest that as fertilizer application on maize increases beyond a threshold level, an exponentially greater proportion of nitrate is leached to the river system before being used by plants, as has been observed some field studies [*Davis et al.*, 2000; *Sogbedji et al.*, 2000]. As a consequence, fertilizer applied in excess of the theoretical threshold has a negligible impact on crop yield and a substantial impact on nitrate export. The magnitude of the threshold will depend on the residual nitrogen level in the soil, as well as soil texture and local weather conditions.

[57] The model also demonstrated that a change in fertilizer application has the greatest impact on maize yields during years with high precipitation and runoff, confirming that accurate forecasting of seasonal weather conditions could help determine the fertilizer application necessary to sustain annual yields. However, at higher levels of fertilizer application, nitrogen leaching becomes increasingly sensitive to the hydrologic conditions, particularly if there is ample residual nitrogen in the soil. Therefore, increasing fertilizer application to counteract losses due to high precipitation will result in a small increase in crop yields, but a disproportionately greater increase in nitrogen loading to the aquatic system.

[58] A decrease in fertilizer application from 1974–1994 would significantly have reduced nitrogen export by the Upper Mississippi Basin, yet the effect may not have been large until the latter part of the time period. If nitrogen storage in the basin continues to increase in the future, this lag time may increase, making it increasingly difficult to decrease nitrate export from the Upper Mississippi Basin. The challenge of reducing nitrate export would be further enhanced if variability in precipitation continues to increase. Any effort to reduce nitrate export from the Upper Mississippi Basin, without significantly affecting crop yields, would therefore have to account for the impact of historical land use practices on nitrogen storage in the soil and groundwater system.

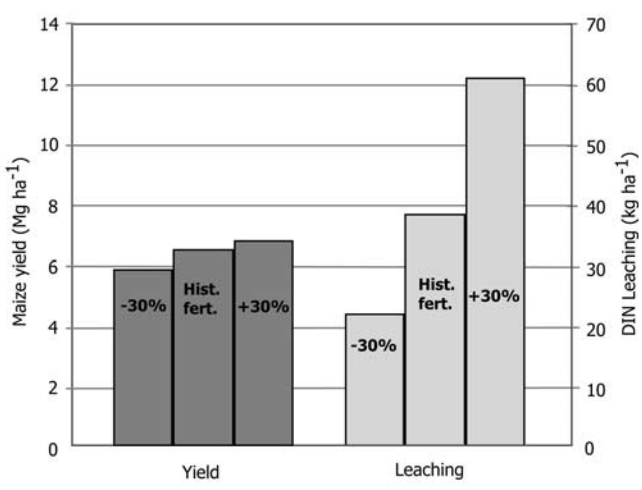


Figure 14. Simulated annual mean maize yield (Mg ha^{-1}) and nitrogen leaching from maize (kg ha^{-1}) for the Upper Mississippi Basin from 1974–1994 under the three fertilizer scenarios. They are plotted at relative scales to illustrate the relative effect of increasing fertilizer use on maize yield and nitrogen leaching.

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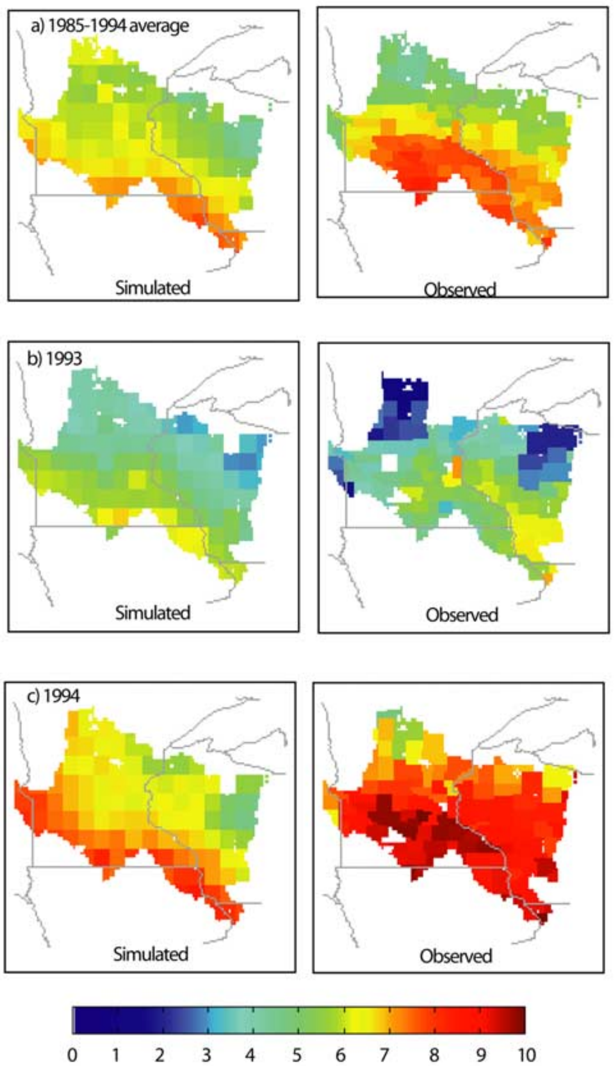

Figure 4. Simulated and observed maize yield (Mg ha^{-1}) for (a) 1985–1994 average, (b) 1993 and (c) 1994. Yield is displayed for each $5' \times 5'$ grid cell with at least 1% soybean cover. Observed maize yield was determined from USDA county-level estimates.

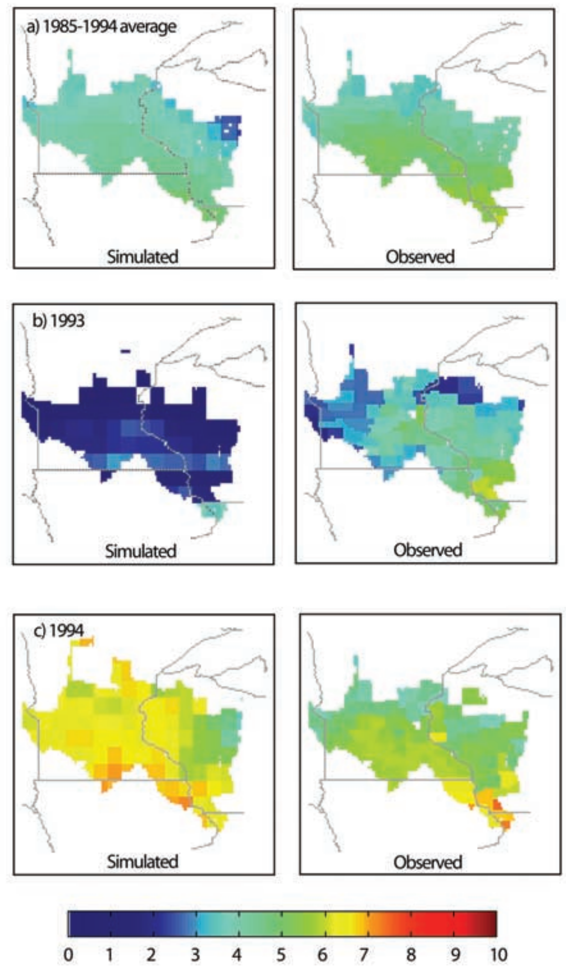

Figure 5. Simulated and observed soybean yield (Mg ha^{-1}) for (a) 1985–1994 average, (b) 1993 and (c) 1994. Yield is displayed for each $5' \times 5'$ grid cell with at least 1% soybean cover. Observed soybean yield was determined from USDA county-level estimates.

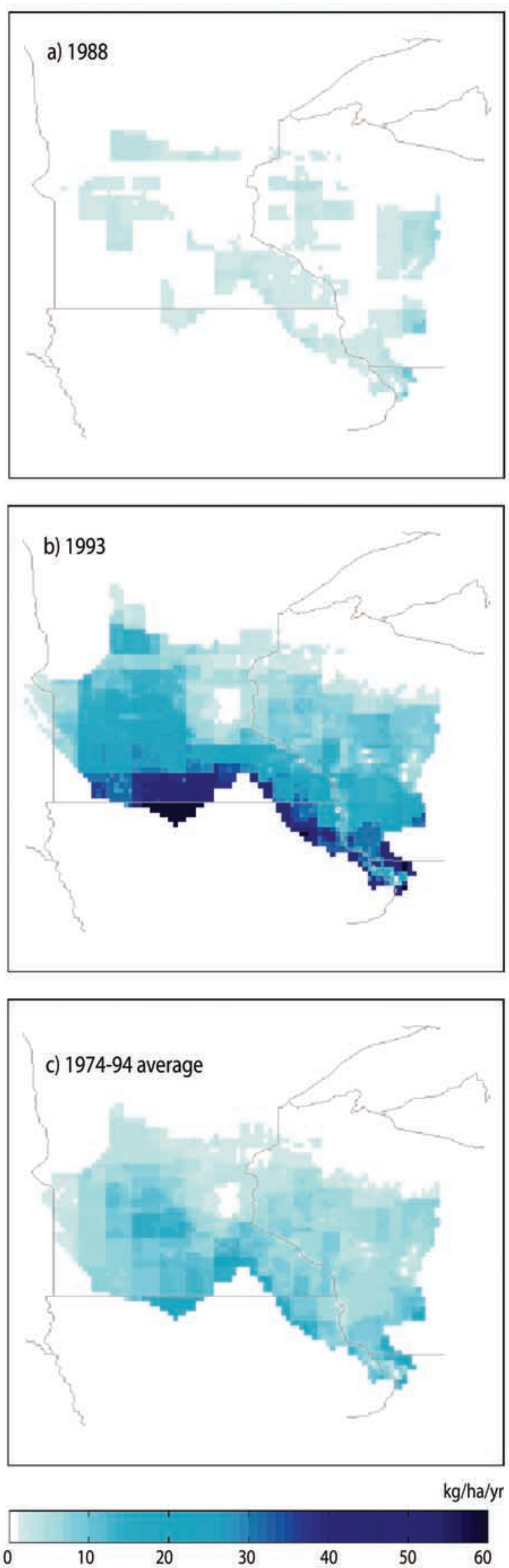

Figure 7. (opposite) Simulated annual dissolved inorganic nitrogen leaching (kg ha^{-1} yr^{-1}) over the Upper Mississippi Basin for (a) 1988, (b) 1993 and (c) 1974–1994 average. Simulated leaching rates from the natural vegetation, soybean and maize control run were integrated with the fractional crop cover data to develop these maps.